The effects of river pulsing on sedimentation in created riparian wetlands

Amanda M. Nahlik and William J. Mitsch

Schiermeier Olentangy River Wetland Research Park, Environmental Science Graduate Program and School of Environment and Natural Resources, The Ohio State University

Introduction

The landscape position of riparian wetland ecosystems, located between terrestrial and riverine systems, leads to runoff from terrestrial systems and floodwaters from aquatic systems (Brown, 1985; Mitsch 1995; Mitsch and Gosselink, 2000), both of which are sources of high sediment loads. Reduced water velocity of wetlands promotes sedimentation, one of the fundamental biogeochemical processes that occurs in wetlands (Mitsch and Gosselink, 2000; Harter and Mitsch, 2003). The process of sedimentation contributes significantly to several wetland functions. Although wetlands provide many ecological services, nutrient cycling is a key function that can result in nutrient retention. Because sediments sorb nutrients, other minerals, and contaminants alike, the process of sedimentation can contribute to nutrient and contaminant removal from the water column. The negative charge of some ions such as phosphates make them particularly susceptible to binding to positively charged ions such as calcium and iron and precipitating as sediments or binding directly to positively charged sites on mineral sediments. Sedimentation also contributes to improved water conditions by increasing water clarity via reduced turbidity. As a result of increased water clarity, more sunlight is able to penetrate the water column, thus potentially increasing aquatic productivity and supplementing dissolved oxygen within a wetland's water column.

Once sediments have settled as substrate, nutrients and contaminants are stored within the wetland, transformed through nutrient cycling, retained or made bioavailable to plants. Some wetlands function as nutrient sinks and have especially high storage capacity for nitrogen and phosphorus, nutrients that have been attributed to the eutrophication of inland lakes and reservoirs and hypoxia in coastal systems, such as the Gulf of Mexico (Rabalais et al., 1999, Scavia et al., 2003).

Pulsing

The flood pulse concept (FPC) encompasses the idea that connectivity of rivers and floodplains through the lateral movement of water (i.e., floods) directly impacts the integrity of both the river and riparian systems (Junk et al., 1989). Although the FPC was originally developed using tropical Amazonian systems as a model, the underlying premise of the FPC has been widely applied to temperate riverine systems (Galat et al., 1998, Sparks et al., 1998, Tockner et al., 2000, Delong et al., 2001, Toth et al., 2002). The hydrologic regime of a wetland and connecting rivers influences an array of ecological processes including sedimentation rates and patterns. Riparian wetlands connected by floodplain to adjacent rivers or streams receive hydrologic pulses during over-bankflow conditions, which generally occur during winter and spring in the Midwestern USA. The exchange of water between rivers and wetlands is essential for the development of both systems (Junk et al., 1989). High volumes of floodwater that periodically overflow into riparian wetlands subsidize the wetland by supplying fresh sediments , providing an influx of nutrients, introducing seeds and organisms, and flushing out old sediments and toxins (Mitsch et al., 1984; Loucks, 1989; Odum et al., 1995, Knighton, 1998; Heimann and Roell, 2000). Likewise, a flush of older wetland sediments can accompany large flood pulses (Davide et al., 2003), providing the river with sediments rich in organic matter and detritus, a critical input to the benthic riverine food chain (i.e., macroinvertebrate; Horne and Goldman, 1994).

Fluctuations in water level and allochthonous inputs provided by a pulsing regime is also essential for several wetland functions, including production, succession, and decomposition (Middleton, 2002). In the Midwestern USA, a natural pulsing regime consisting of flooding in winter and spring and a drier period in summer and autumn allows for effective dispersal, introduction, and germination of seeds into riparian wetlands (Middleton, 2000). Robertson et al. (2001) and Darke and Megonigal (2003) both reported that the spring timing of flood pulses significantly increased biomass production and species richness in riparian forested wetlands. Disturbances caused by rapid changes in water levels and velocities that accompany pulses help drive succession by creating open niches and introducing new species into the system (Parsons et al., 2005). The amount of moisture in a wetland can affect decomposition rates such that the wettest areas have the highest decomposition rates (Brinson, 1977), although some studies have also shown that continuous anaerobic conditions do not support microbes responsible for decomposition (Reddy and Patrick 1975). Brinson et al. (1981) suggest that the most effective hydrologic regime for high decomposition is a pulsed system with dry periods in between wet periods.

Goals and objectives

The ability of scientists to quantify sedimentation rates and patterns, particularly during pulsing events, is important for predicting wetland succession and viability, land accretion, and the efficiency nutrient removal by wetlands. In addition, there have been relatively few studies on sedimentation conducted in freshwater wetlands compared to saltwater marshes with the exception of Mitsch (1979), Brueske and Barrett (1994), Fennessy et al. (1994), Kleiss (1996), Wardrop and Brooks (1998), Craft and Casey (2000), Liptak (2000), Mann and Wetzel (2000), Sánchez-Carrillo et al. (2001), and Harter and Mitsch (2003). Understanding sedimentation and being able to better predict its effect on short-term and long-term processes of a wetland will allow for better wetland design and placement so that goals for water quality and habitat creation can be met.

The goal of this project was to determine how hydrologic pulsing in two created riparian wetlands affects rates and patterns of sedimentation and accompanying nutrient dynamics. It was hypothesized that sedimentation would be higher in pulsed rather than steady-flow conditions, due to both more rapid sedimentation and subsidized autochthonous inputs (i.e., aquatic productivity). This goal was investigated by the following specific studies in two full-scale (1-ha) experimental wetlands:

1) Comparison of sedimentation rates during non-growing and growing seasons in both pulsed and steady-flow years;

2) Investigation of organic sedimentation as an indicator of the importance of autochthonous productivity in riparian wetlands;

3) Quantification of the importance of pulsing in nutrient sedimentation and subsequent retention.

Methods

Site description

Two 1-ha experimental wetlands located at The Olentangy River Wetland Research Park (ORWRP) in Columbus, Ohio, U.S.A. (latitude 40.021°N, longitude 83.017°E) were used in this study. The experimental wetlands were excavated in 1993 on floodplain previously farmed by The Ohio State University. A water intake system allows both hydrologic control of the system (i.e., water input can be manipulated) and parity between the two wetlands (i.e., the two wetlands always receive the same amount of water). One wetland was planted with 13 species of hydrophytes in 1994 and the other left to natural colonization (Mitsch et al., 1998, 2005b, c).

Experimental design

In 2004, the two experimental wetlands were subjected to pumped high-flow pulses for the first week of the first six months (January through June), during the typically wet season in the Midwestern USA. Mean inflow rates from the river to the wetland was increased to an average of 52 ± 3 cm d^{-1} for seven days, after which flow rates were reduced to a mean low steady-flow rate of 7 ± 0.2 cm d^{-1} (Figure 2). During the last six months of 2004 (July through December), during the typically dry season, a mean moderate-flow rate of 10 ± 0.01 cm d^{-1} was maintained. In 2005, the wetlands received a steady-flow regime for the duration of the year, where water was pumped into the wetlands at a constant rate of 11 ± 0.01 cm d^{-1} with no major hydrologic fluctuations (i.e., no pulses). To reduce experimental variability, the total hydrologic input into the wetlands in 2005 was designed to be equivalent to that in 2004. Hydrologic loading rates take into account the area of the wetland and are reported as $m^3 m^{-2} d^{-1}$ or m d^{-1}.

Sediment traps

Sediment traps were constructed with two 500 ml wide-mouth Nalgene bottles attached to each other by 0.5-cm rubber bands and wire and secured to metal rebar driven deep into wetland sediments using a modified design from Fennessy et al. (1994). The two bottles used for each trap had a height:diameter ratio >3 to minimize resuspension of trapped sediments (Hackanson and Jansson, 1983). The bottles were buried in the substrate so that the opening was about 5 cm above the soil surface; this biased for new or resuspended sediments falling from the water column as opposed to older sediments that are rolled, slid, or saltated across the soil surface via bed-material transport (Gardner, 1980), and allowed traps to function during low flows. To minimize trapping sediments disturbed during placement, sediment traps were filled with water and capped before deployment. After fixing the traps to the rebar, a minimum

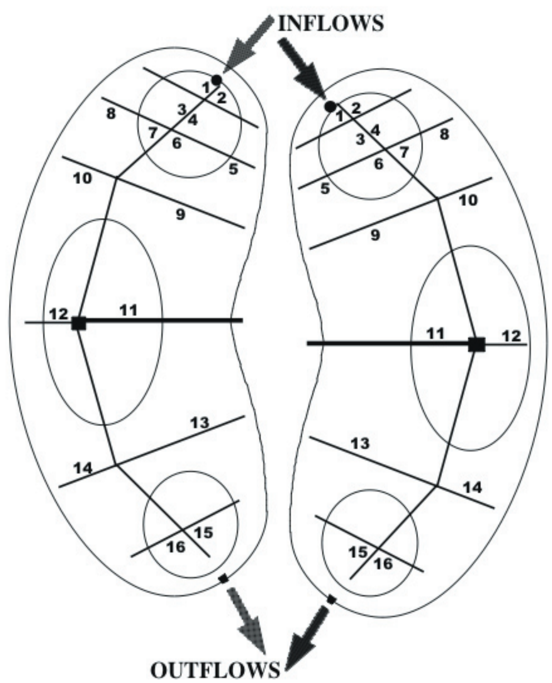

Figure 1. Location of sixteen sediment trap stations in each experimental wetland at the Olentangy River Wetland Research Park. Deep marsh areas (generally >30 cm deep) are approximated by the circles within the basins. Lines represent boardwalks.

of 30 minutes was allowed for any disturbed sediments to settle before lids were removed from the bottles.

Sediment traps were deployed on the same schedule: once per month in April, May, June, and July of 2004 and 2005. Experimental months, consisting of 2004 pulsed and 2005 steady-flow regimes, included April, May, and June. Pulses lasted for eight days during each of these months, with the remaining sampling time under a steady-flow regime. July was subjected to a steady-flow regime in both 2004 and 2005.

Sediment traps were placed at 16 sites in each of the two wetlands each month (Figure 1). Because previous studies have shown that most sediments settle near the inflow (Hakanson and Jansson, 1983, Brueske and Barrett, 1994; Fennessy et al., 1994; Kleiss, 1996; Sánchez-Carrillo et al., 2001; Zierholz et al., 2001), ten sites were clustered near the inflow of the wetland and the remaining six were spaced at wider intervals towards the outflow. Traps were deployed in both deep and shallow areas of each wetland: stations 1–7, 11–12, and 15–16 were placed in deep areas, while stations 5, 8, 9–10, and 13–14 were placed in shallow areas of the wetlands.

Sediment traps were retrieved monthly and new ones were deployed at the same time. Before retrieving traps from the wetland, caps were placed back onto the submerged bottles to minimize sediment loss during removal. Bottles were labeled upon collection and stored in a cooler at 4°C until analysis.

Laboratory analysis

Sediment trap processing and analysis began between 2–14 days after collection to allow sediments to resettle in bottles after transport from the field to the lab. After sediments were resettled and the water layer was relatively free from suspended sediments, supernatant was poured with caution to avoid disturbing the sediments. After the majority of the supernatant layer was removed, the sediments were poured into pre-weighed aluminum drying pans. Any remaining sediments in the bottles were rinsed into the drying pans using distilled water. Sediments were dried in a drying oven at 60–65°C until sediments reached constant weight, cooled in a desiccator for a minimum of 12 hours, and weighed with a top-loading balance (Mettler AE200) up to 0.0001 g.

Because standard methods for total suspended solids call for drying temperature of 103–105°C (APHA, 1998), and collected sediments were dried at 60–65°C to preserve the integrity of nutrients (Comin et al., 1997, Verhoeven et al., 2001), sub-samples of collected sediments were dried at 103–105°C. Sedimentation rates were corrected for the difference in soil weights using a linear regression equation, and reported as dried at 103–105°C. Sedimentation rates,

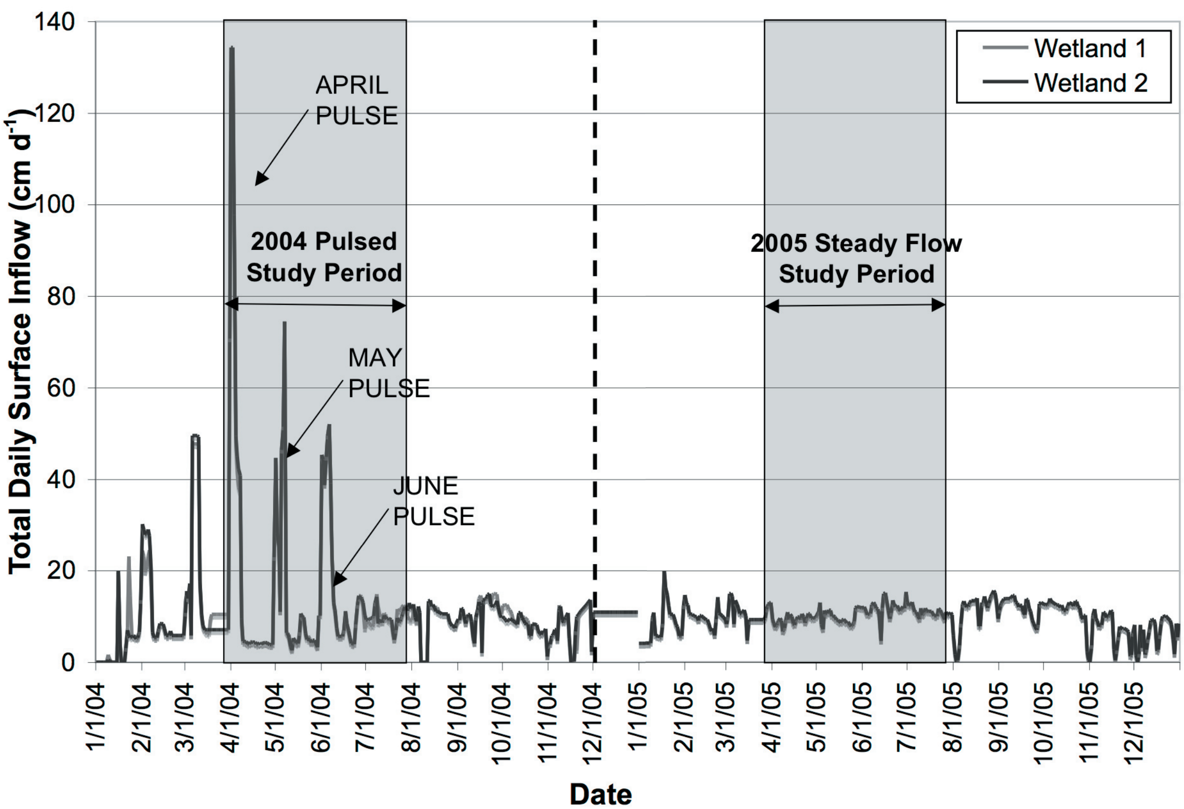

Figure 2. Total surface water inflow (cm d^{-1}) in experimental Wetland 1 and Wetland 2 for 2004 pulsed and 2005 steady-flow years. Shaded boxes indicate periods for this study. Monthly pulses in 2004 are designated with arrows.

determined as the average of the two sediment traps at each site, are reported in g-dry wt m^{-2} mo^{-1}.

Dried samples were ground to pass through a 2-mm sieve and stored in airtight plastic bags until loss on ignition (LOI) and nutrient analyses. To determine the organic content of sediments, LOI tests were performed. Sub-samples of approximately 5 g were placed into porcelain crucibles of determined weight and combusted at 550°C for approximately three hours in a Fisher Scientific Isotemp® Programmable muffle furnace (650–750 Series) (Davies, 1974; Forster, 1995). After combustion, samples were cooled to room temperature in a desiccator and weighed to determine combusted weight. Percent organic matter (% OM) was calculated, and organic matter sedimentation rates are reported in g-dry wt m^{-2} mo^{-1}. Inorganic material sedimentation was determined by the difference between total sedimentation rates (inorganic and organic sediments) and organic matter sedimentation rates.

Nutrient analyses were performed for deep-water sites at the inflow (Sites 6 and 7), middle (Sites 11 and 12), and outflow (Sites 15 and 16) of each wetland for each sampling month in 2004 and 2005. Prepared samples were analyzed by the Ohio Agricultural Research and Development Center (OARDC) Service Testing and Research (STAR) Laboratory at The Ohio State University Wooster Campus. Soils were analyzed for total N using the Dumas Method combustion technique (AOAC 2002). Mineral analysis for P and Ca was performed by inductively coupled plasma (ICP) emission spectrometry after 3051 Microwave Digestion (U.S. EPA, 1994).

Turbidity

Daily water samples were collected from the inflow and outflows of the wetlands in the morning and evening throughout this study, and river water samples were collected on a weekly basis near the water intake pump for the wetlands. Samples were refrigerated at 4°C upon reaching the laboratory and later analyzed for turbidity with a Hach 2100N Turbidimeter using nephelmetric methods (APHA, 1998).

Total suspended solids for the water samples collected during the study period in river, inflows, and outflows were estimated using turbidity data and a regression equation reported in Harter and Mitsch (2003).

Data analysis

Databases were made using Microsoft Excel. Graphs and figures were made in Microsoft Excel and Adobe Illustrator CS. Minitab 14 was used to conduct analysis of variance (ANOVA), Fishers Pairwise Comparisons, t-tests, and Pearson Correlations. ANOVA tests were used to compare temporal and spatial sedimentation. Fishers Pairwise Comparisons were conducted at 95% confidence interval to determine differences between months and indicate $p \leq 0.05$. T-tests were used to compare July 2004 and July 2005 to other monthly data. Significant differences indicate $p \leq 0.05$.

Results

Hydrology and river sediments

This study compared hydrologic pulses in April, May, and June 2004 with steady flow conditions maintained from over a similar period in 2005 (Figure 2). Steady flow conditions in July of both years, representative of post-experiment conditions, were included in the comparison. Neither hydrology nor sedimentation patterns were significantly different between the two experimental wetlands; therefore, they were treated as replicates throughout this study. Total surface inflow (averaged between the wetlands) for the study period (April through July was 20 m in 2004 and 13 m in 2005 with a significant difference between the two years ($p = 0.011$)). Statistical analyses revealed a weak significant difference between the total suspended solids in the river between 2004 and 2005 (means of 14 ± 1 and 10 ± 1 mg L^{-1}; $p = 0.049$; Table 1)). Total suspended solids were also significantly different in the inflows of the wetlands between 2004 and 2005 with reported means of 11 ± 1 and 7 ± 1 mg L^{-1}, respectively ($p = 0.001$).

Sedimentation rates

Sediments were collected monthly over a period of 126 days in both 2004 and 2005. Total sedimentation for the study period (April, May, June, July) was 90 kg m^{-2} in 2004 and 64 kg m^{-2} in 2005, with significantly higher means in 2004 than 2005 ($p = 0.032$). When only taking into account experimental months (April, May, June), total sedimentation was not different between the pulsed months in 2004 (45 kg m^{-2}) and corresponding steady-flow months in 2005 (39 kg m^{-2}). There was higher total sedimentation during the 2004 study period than the 2005 study period because of significantly higher sedimentation rates in July 2004 than in July 2005 (Figure 3a; $p = 0.011$).

Inorganic/organic material sedimentation

Sedimentation was dominated by inorganic material throughout the study (Figure 3a). There were significant differences in monthly inorganic material sedimentation in pulsed and steady flow years ($p_{2004} = 0.006$; $p_{2005} = 0.002$). Sedimentation rates for inorganic material did not statistically change between April and May of the pulsed and steady-flow years. In June, however, inorganic material sedimentation significantly increased from May by 52 and 56% in the 2004 pulsed and 2005 steady-flow years, respectively ($p \leq 0.05$). While there were increases in inorganic material sedimentation from the previous month, there was no difference in rates between June 2004 and June 2005. Inorganic material sedimentation significantly increased in the steady-flow July 2004, with a 61% increase from the pulsed June 2004 ($p = 0.020$). There were no significant increases in inorganic material sedimentation from June to July in the 2005 steady-flow year. Inorganic material sedimentation was significantly different between July 2004 and July 2005, despite steady-flow conditions in July for both years ($p = 0.006$).

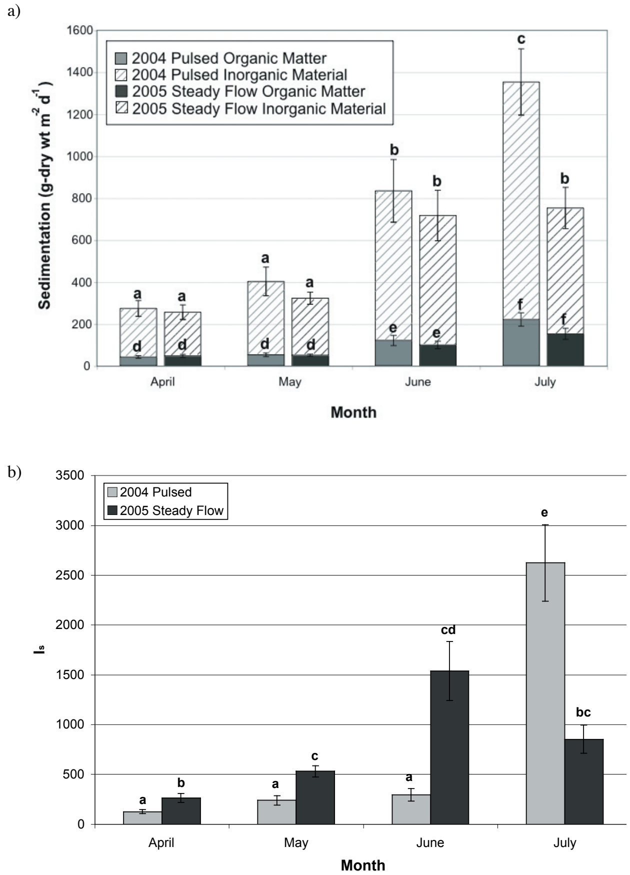

Figure 3. Sedimentation rates a) separated by organic matter sedimentation (solid bars, bottom) and inorganic material sedimentation (striped bars, top) and b) reported as monthly Is, or sedimentation index, in 2004 pulsed and 2005 steady-flow years). Total sedimentation is reported as g-dry wt m^{-2} d^{-1} and the sedimentation index is unitless. Means are reported with standard error. n-values for 2004 are 32, 32, 31, and 29 for each month, respectively. n-values for 2005 are 32 for each month. Different letters signify significant differences at $p \leq 0.05$.

Organic matter comprised a small fraction of the total sedimentation throughout the study (Figure 3a) and there were significant differences in monthly organic matter sedimentation between pulsed and steady flow regimes ($p_{2004} < 0.001$; $p_{2005} = < 0.001$). Sedimentation rates for organic matter did not differ between April and May of the pulsed and steady-flow years. Organic matter sedimentation rates did significantly increase from May to June in the pulsed and steady-flow years by 50 and 46%, respectively ($p_{2004} = 0.031$, $p_{2005} = 0.044$), although rates in June 2004 and June 2005 were not different. Organic matter sedimentation rates significantly increased from June to July, with a 45 and 35% increase in 2004 and 2005, respectively ($p_{2004} = 0.004$; $p_{2005} = 0.030$). There was no difference in organic matter sedimentation between July 2004 and July 2005.

Normalized sedimentation rates

To compare sedimentation rates with monthly differences in imported total suspended solids and inflow rates into the wetlands between years, a sedimentation index (I_s) value was calculated each month using the following equation:

$$I_s = s / (q \times Ci) \qquad (1)$$

where, s is equal to sedimentation rates (g m^{-2} mo^{-1}), q is equal to the total water input into the wetland (m mo^{-1}), Ci is equal to the mean concentration of total suspended solids imported into the wetland (mg L^{-1} = g m^{-3}). Because all units cancel, I_s is unitless and provides a relative comparison between months and years.

Total normalized sedimentation (I_s) was not significantly different between study periods (April, May, June, July) in 2004 and 2005; however, I_s for the 3-month period April, May, and June was significantly lower in the pulsing year than in the corresponding steady-flow months ($p_{April} = 0.002$, $p_{May} = 0.008$, $p_{June} = 0.010$). There were no differences among April, May, and June 2004 sedimentation indices, but there was an eight-fold increase in I_s from June to July 2004 ($p < 0.001$; Figure 3b). April and May 2005 did not have significantly different I_s values, but June was significantly higher than the other months, including July ($p = 0.005$). Differences between 2004 and 2005 occurred in the months of April, May, June, and July, with steady-flow years exhibiting significantly higher I_s values ($p_{April} = 0.002$, $p_{May} = 0.008$, $p_{June} = 0.010$, $p_{July} < 0.001$). July showed an 85% decrease between 2004 and 2005.

Spatial patterns

Water depth had no effect on sedimentation, although sedimentation rates were generally higher within the first half (inflow) of the wetlands than those sites beyond the middle of the wetland, towards the outflow (Figure 4; $p = 0.003$). Sedimentation patterns varied with distance from the inflow, with similar patterns in the pulsed and steady-flow years. Significant differences between the study years occurred at 110 and 140 m from the inflow ($p_{110} = 0.047$; $p_{140} = 0.008$). Peaks in sedimentation rates above 600 g-dry wt m^{-2} d^{-1} occurred at 5, 45, and 140 m from the inflow during the pulsed year, while peaks during the steady-flow year were only present at 45 and 110 m from the inflow. The wetland cross-section shown in Figure 4 suggests that sedimentation peaks at 45 and 110 m may coincide with shallow, normally vegetated areas within the wetlands, while lower sedimentation rates generally correspond to the deep, open water basins.

Sediment retention

Sediment retention data was gathered from daily water samples taken at inflows and outflows; means normalized for flow were determined for pulsed weeks in 2004, steady-flow weeks in 2004, and steady-flow in 2005. There was higher total sediment retention in the wetlands during the pulsed study period (465 kg) than during the steady flow period (163 kg). Assuming the average inflow TSS concentrations reported earlier (Table 1), total sediment retention expressed as percent of sediment inflow was 33.5 and 18.4% for 2004 and 2005, respectively.

Sediment retention was higher in pulsed months of April, May, and June (means of 0.6, 0.5, and 0.8 g m^{-2} d^{-1}, respectively), while the same months during the steady-flow months had lower sediment retention (means of -0.1, 0.1, and 0.1 g m^{-2} d^{-1}, respectively). In July 2004, there was a net export of sediments after the pulsing occurred (mean of -0.4 g m^{-2} d^{-1}), while in July 2005, the highest retention of sediments after three continuous months of steady flow occurred (mean of 0.4 g m^{-2} d^{-1}).

Organic matter and inorganic material

Percent organic matter of collected sediments was consistently higher in the 2005 steady-flow year than in the 2004 pulsed year with means of 18 and 15%, respectively ($p < 0.001$). June was the only month that was not significantly different between 2004 and 2005 for both organic and inorganic material. Both years exhibited similar temporal patterns of percent organic matter in the sediments, with the greatest percentages of organic matter in April (means of 16 and 20% for 2004 and 2005, respectively) and July (means of 17 and 21% for 2004 and 2005, respectively). Organic content of sediments decreased in both May to 14 and 16% (2004 and 2005) and June to 15% (both 2004 and 2005). Inorganic material showed the inverse pattern, with highest concentrations in May and June and lower concentrations in April and July. This pattern is consistent with results by Tuttle (2005), who found generally higher aquatic productivity in the water column in the steady-flow year.

Nitrogen

Total nitrogen concentrations of sediments varied from month to month between hydrologic treatments and showed no consistent temporal pattern between pulsed and steady-flow years. Total nitrogen of the collected sediments averaged 0.50 and 0.49% in April and May (respectively) of 2004, and decreased to 0.41% in both June and July of the same year ($p = 0.002$). The steady-flow year did not change significantly between April, May, and June (0.47, 0.44, 0.47%, respectively), but increased by 15% to 0.6% nitrogen from June to July ($p = 0.014$). Significant differences in total nitrogen content of sediments were not present between years in April; however, differences in May, June, and July were evident. The pulsed year nitrogen content was lower in May 2004 than in May 2005 ($p = 0.055$), while the steady-flow year exhibited higher nitrogen concentrations in June and July from the pulsed year ($p_{June} = 0.045$; $p_{July} = 0.007$).

Phosphorus

Neither year nor month had a significant effect on phosphorus content of sediments. Monthly phosphorus concentrations were consistent within the range of 975 to 1070 μg/g. Although there were no significant differences in phosphorus contents, they tended to follow the same temporal pattern as total nitrogen content of sediments.

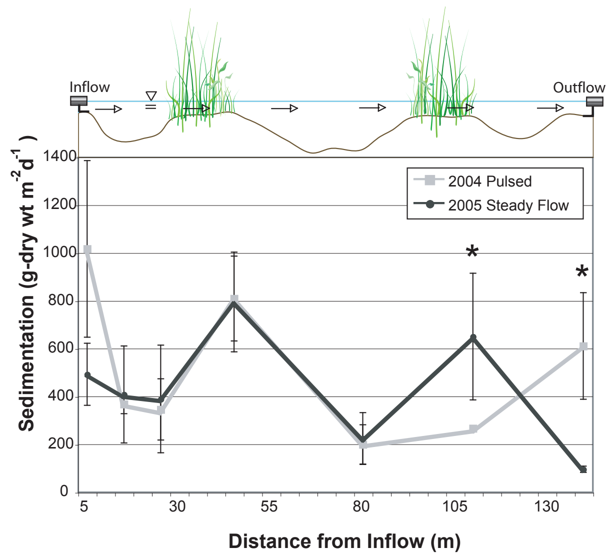

Figure 4. Sedimentation rates (g-dry wt m^{-2} d^{-1}) as a function of distance from the inflow (m) in 2004 pulsed and 2005 steady-flow years. Means are reported with standard error. Asterisks (*) represent significant differences between pulsed and steady-flow years at $p \leq 0.05$. Cross-section of the wetlands approximates the basin morphology and ecology, including basin depth and presence of vegetation, at corresponding distances.

Calcium

The calcium content of captured sediment was higher overall in the steady-flow year than the pulsed year, and significant differences between years occurred in June and July ($p_{June} < 0.001$; $p_{July} = 0.002$; Figure 5a). Steady flow calcium content increased between 2004 and 2005 by 250% in June and 82% in July. The pulsed year showed little variation in calcium content from month to month; only June had significantly lower calcium content than the other months with a mean of 29.7 mg/g ($p = 0.009$). There were no significant differences between calcium content of sediments from April and May of the steady-flow year. June and July of 2005 had significantly higher calcium concentrations than earlier months ($p < 0.001$). Calcium concentrations nearly doubled from May to June 2005 and remained high in July 2005.

Nutrient sedimentation

Mean nitrogen and phosphorus sedimentation did not differ between years (Table 2). In 2004, nitrogen sedimentation increased from the inflow to the middle to the outflow of the wetlands (means of 1.8, 2.3, 2.9 g-N m^{-2} d^{-1}), while the opposite pattern occurred in the steady-flow year (means of 1.9, 1.1, and 0.5 g-N m^{-2} d^{-1} for the inflow, middle, and outflow, respectively); however, these spatial patterns are only trends and are not statistically different at $p \leq 0.05$ ($p = 0.096$). Higher phosphorus sedimentation rates occurred in the pulsing year from the inflow to the middle to the outflow with means of 0.4, 0.5, and 0.6 g-P m^{-2} d^{-1}. Conversely, phosphorus sedimentation

Table 1. Mean total suspended solids (TSS) (mg L^{-1}) estimated from turbidity analyses in the Olentangy River and inflow to the experimental wetlands (always measured at Wetland 1) during both years of experimental months. Monthly averages are reported with standard error. n-values are in parentheses.

	River		Wetland Inflow	
Month	2004	2005	2004	2005
April	14 ± 3 (9)	15 ± 3 (5)	10 ± 1 (25)	10 ± 1 (26)
May	13 ± 2 (7)	9 ± 3 (4)	12 ± 1 (23)	6 ± 1 (25)
June	19 ± 3 (6)	7 ± 0 (3)	16 ± 2 (23)	4 ± 0 (19)
July	3 ± na (1)	7 ± 1 (4)	5 ± 0 (19)	8 ± 1 (17)
Mean	14 ± 1 (23)	10 ± 1 (16)	11 ± 1 (90)	7 ± 1 (87)

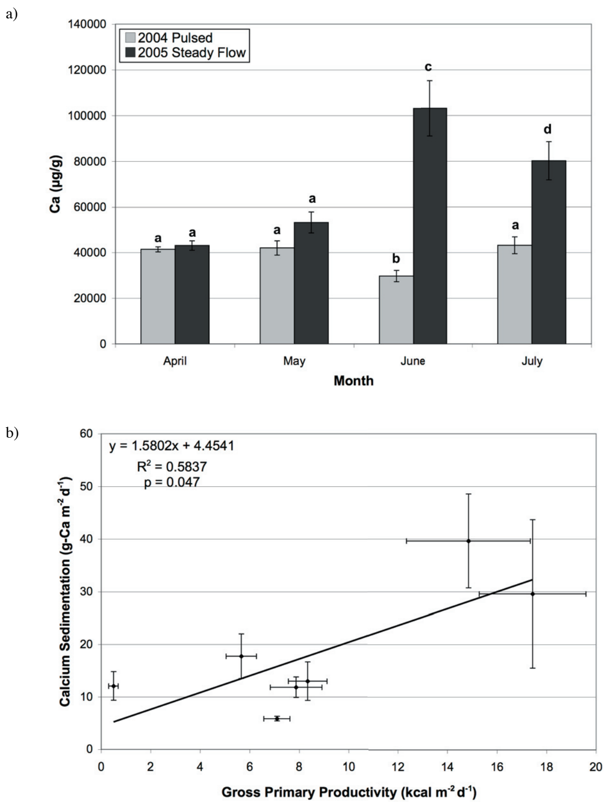

Figure 5. a) Mean concentrations of calcium (µg/g) of trapped sediments in 2004 pulsed and 2005 steady-flow years. Means are reported with standard error. n-values are 9, 11, 11, 9 in 2004 and 12, 12, 10, 12 in 2005 for April, May, June, and July (respectively). Different letters signify significant differences at $p \leq 0.05$. b) Calcium sedimentation (g-Ca m^{-2} d^{-1}) as a function of gross primary productivity in the water column (kcal m^{-2} d^{-1}). Error bars represent standard error. P-value determined using a Pearson Correlation. Gross primary productivity data provided by Tuttle (2005).

rates decreased with distance from the outflow during the steady-flow year (0.6, 0.2, and 0.1 g-P m^{-2} d^{-1} for inflow, middle, and outflow, respectively). Significant differences in phosphorus sedimentation between years occurred at the outflow ($p = 0.001$).

Calcium sedimentation rates were generally higher in the pulsed year than the steady-flow year (Table 2); however, only calcium sedimentation rates for the outflow were

Table 2. Sedimentation rates for nitrogen (g-N m^{-2} d^{-1}), phosphorus (g-P m^{-2} d^{-1}), and calcium (g-Ca m^{-2} d^{-1}) during the study period in 2004 pulsed and 2005 steady-flow years. Means are reported with standard error.

	2004 Pulsed	2005 Steady Flow
Nitrogen	2.3 ± 0.5	1.2 ± 0.4
Phosphorus	0.51 ± 0.11	0.29 ± 0.10
Calcium	21 ± 5	17 ± 5

significantly higher in 2004 than 2005 ($p = 0.005$). The same trend of increased rates in the pulsed year and decreased rates in the steady-flow year from inflow to outflow exhibited by nitrogen and phosphorus sedimentation also applied to calcium sedimentation. Mean calcium sedimentation increased from 16, 20, 26 g-Ca m^{-2} d^{-1} in the inflow, middle, and outflow (respectively) during the pulsed year and 27, 15, and 9 g-Ca m^{-2} d^{-1} in the inflow, middle, and outflow (respectively) during the steady-flow year.

Discussion

Effects of pulsing on sedimentation

Hydrologic pulsing appears to have little effect on sedimentation rates in the riparian wetlands studied here, as there was no difference in uncorrected and normalized sedimentation rates between experimental pulsed months (April, May, June) and corresponding steady flow months. Cumulative sediment accumulation during the 4-month study period that included a post-flooding month (April, May, June, July) was significantly higher in 2004 than 2005, with an estimated 900,000 ± 15,000 kg of deposited sediments in 2004, compared to 640,000 ± 10,000 kg in 2005 in the experimental wetlands. Similar I_s values between the two years suggest that when sedimentation rates were corrected for differences in flow and total suspended solids, monthly patterns of sedimentation rates varied depending on the hydrologic regime, but the same amount of sediments were deposited during the study.

Mean sediment retention in the wetlands, as estimated from water quality measurements at the inflow and outflow, was significantly higher in the 2004 pulsing period than

2005 steady-flow period (means of 90 and 64 kg for 2004 and 2005, respectively). The highest percent retention was found to coincide with the actual pulse (i.e., the eight pulsed days), while the subsequent steady-flow period (i.e., the remaining 20–25 days) generally had a net export of sediments from the wetland. Sediment flushing after pulses has been reported by others (Loucks, 1989). We believe that new sediments imported from the river are retained and distributed throughout the wetlands, while older sediments are exported. This process of sediment exchange is essential for the wetland by increasing nutrient sorption potential and maintaining stable depths. Exported organic sediments contribute to benthic food webs in adjacent rivers (Horne and Goldman, 1994).

Allochthonous v. autochthonous sediments

Determining the dominant source of sediments and making quantitative estimates on the sediment input from a particular source can be difficult without stable isotope studies, e.g., Gichuki et al., 2001; Kendall et al., 2001; Neumann et al., 2002; Ziegler and Fogel, 2003); however, sediment characteristics we measured in this study can suggest the relative importance of the two sources. The ratio of organic matter to inorganic material and nutrient concentrations of collected sediments were used as indicators of the particulate material source.

While some studies on riparian systems have reported that allochthonous sedimentation rates exceed those of autochthonous (organic) sedimentation (Sánchez-Carrillo et al., 2001), others have shown that riparian wetlands can be dominated by autochthonous sedimentation, unless there is a natural river pulse, which contributes high amounts of inorganic material (Mitsch et al., 1979). Mean inorganic sedimentation rates were higher in the pulsed year than those in the steady-flow year, although some of this difference may be due to the estimated 29% lower sediment load during the steady-flow year (Figure 3a). Months that experienced long periods of steady flow after pulsing, such as July 2004, had significantly higher inorganic sedimentation rates than the steady-flow year. Organic sedimentation rates did not change significantly, although monthly means were consistently higher in the pulsed year. The current study showed that sedimentation in riparian wetlands is dominated by allochthonous inputs, regardless of hydrologic regime (Figure 3a). This is due to the fact that sediments transported by rivers, such as the Olentangy River, are principally inorganic (Canfield, 1997), with organic matter quickly decomposed or consumed and incorporated into the benthic invertebrate food chain (Vannote et al., 1980). Wetlands fed predominantly by river water will have a greater flux of inorganic sediments. This study showed that inorganic sediments were 82–85% of the collected sediments. The importance of autochthonous sedimentation in the experimental wetlands is evident by comparing organic:inorganic material ratios. The organic: inorganic material ratios of the sediments throughout this study were consistently well below one, indicating that sediments were dominated by allochthonous inputs. Furthermore, autochthonous sources of sediments were more important in the steady-flow year than the pulsing year for three of the four months compared.

Nutrient concentrations of sediments may also help identify their source. As a result of natural fluvial processes, sediments carried by rivers undergo shearing, exposing fresh surface areas void of nutrients (Schumm and Stevens, 1973; Knighton, 1998); therefore, low nutrient concentrations in sediments, especially phosphorus, may indicate that the river is supplying the majority of the sediments. Phosphorus concentrations did not significantly differ between months or years; however, phosphorus concentrations did follow a similar pattern to nitrogen concentrations, which did have significant differences between months and years. Total nitrogen concentrations were higher in the pulsed year for April and May, while the pattern reversed in June and July, with the steady-flow year exhibiting significantly higher nitrogen concentrations. This pattern, found in the nitrogen concentrations and phosphorus concentrations over the study periods, may indicate that nutrients were being contributed either directly from the river water in the early part of the growing season or the higher nutrients in the early part of the season were due to resuspension of wetland sediments. Conversely, low nutrient concentrations in the latter part of the growing season may indicate that much of the sediment is being contributed from the river during the latter months of the pulsing year, while high nutrient concentrations during the latter part of the steady-flow year indicate that sediments are either resuspended or produced within the wetlands via autochthonous pathways.

Comparison to other studies

Sedimentation is affected by many outside factors, such as flow rates, water source, vegetative cover, wind speeds, and animal activity. For this reason, comparing sedimentation rates between wetlands is difficult, and reported sedimentation rates vary greatly in the literature. The range of sedimentation rates is between 1.3 ± 0.2 g m^{-2} d^{-1} in bottomland hardwood forests using sediment discs (Kleiss, 1996) to 98 ± 16 g m^{-2} d^{-1} in the experimental wetlands studied here using soil horizon markers (Harter and Mitsch, 2003). Sedimentation rates in this study for both pulsed and steady-flow years (430–500 g m^{-2} d^{-1}) were substantially higher than other reported rates.

The bottle trap method is a good indicator of relative sedimentation rates, but over-estimates sediment retention (Kleiss, 1996; Steiger et al., 2003). Sedimentation rates determined in this study were higher than most other studies; however, the flow-through rates at which wetlands were subjected during this study were higher than any other study examined (Figure 6). It was expected that sedimentation rates would increase logarithmically with increased flow rates because sediment transport physics rule that higher water velocities have the ability to carry more sediments and sediments of a larger particle size (Knighton, 1998). When the velocity of water is suddenly decreased, such as when floodwaters reach wetlands, the sediment loads settle from the water column, the largest particles dropping first, followed by sediments of decreasing particle sizes (Knighton, 1998). A regression of sedimentation rates at different flow rates for studies that used the bottle trap method in riparian marshes revealed that sedimentation rates in riparian wetlands logarithmically increase with flow rate ($R^2 = 0.8228$). Taking into consideration the high flow rates of this study, calculated sedimentation rates did not overestimate more than other studies using the similar methods.

Sedimentation rates for inorganic and organic material were substantially higher than other freshwater wetland studies; however, the rate at which we introduced water into the experimental wetlands was also considerably higher than other studies. This study on riparian wetlands is more comparable to studies conducted on coastal marshes, such as those along the Mississippi River in Louisiana, where major fluxes of river and marine sediments often occur. The wetlands in this experiment and deltaic marshes share similar water source and pulsing characteristics (Hatton et al., 1983; Boumans and Day, 1994; DeLaune et al., 2003). Despite similar characteristics, sedimentation rates are still higher in this study than in deltaic marshes. Boumans and Day (1994) reported flow rates in deltaic marshes that were nearly twice those in this study; however, sedimentation rates were two orders of magnitude lower. Other deltaic studies report rates of sedimentation similar to the Boumans and Day study (Hatton et al., 1983; DeLaune et al., 2003). Under-estimation of sedimentation rates is probably due to the use of sediment plate methods used (Boumans and Day, 1994).

Sediment budgets

Total suspended solids (TSS) were slightly higher in the pulsed year in both the river and inflow to the wetlands. The higher import of TSS from the river may explain some of the higher sedimentation rates during the pulsing year, especially those for inorganic material, which we expect the river to predominantly contribute. After factoring in monthly flow rates, the total difference between TSS loads in the two years is 127 g m^{-2} mo^{-1}, 56% lower in the 2005 steady-flow year (Table 3). Highest TSS loads were imported in April and June of the pulsed year (73 and 79 g m^{-2} mo^{-1}, respectively) and April and July of the steady-flow year (32 and 29 g m^{-2} mo^{-1}, respectively). These high TSS loads do not, however, correspond to months with the highest sedimentation rates, indicating that the difference in TSS loads into the wetlands between years is not driving sedimentation rates.

The ratios reported in Table 3 for the experimental wetlands are sedimentation:inflow sediment (S:I) and sedimentation:outflow sediment (S:O). S:I over the 4-month period was similar in pulsed and steady-flow years. July

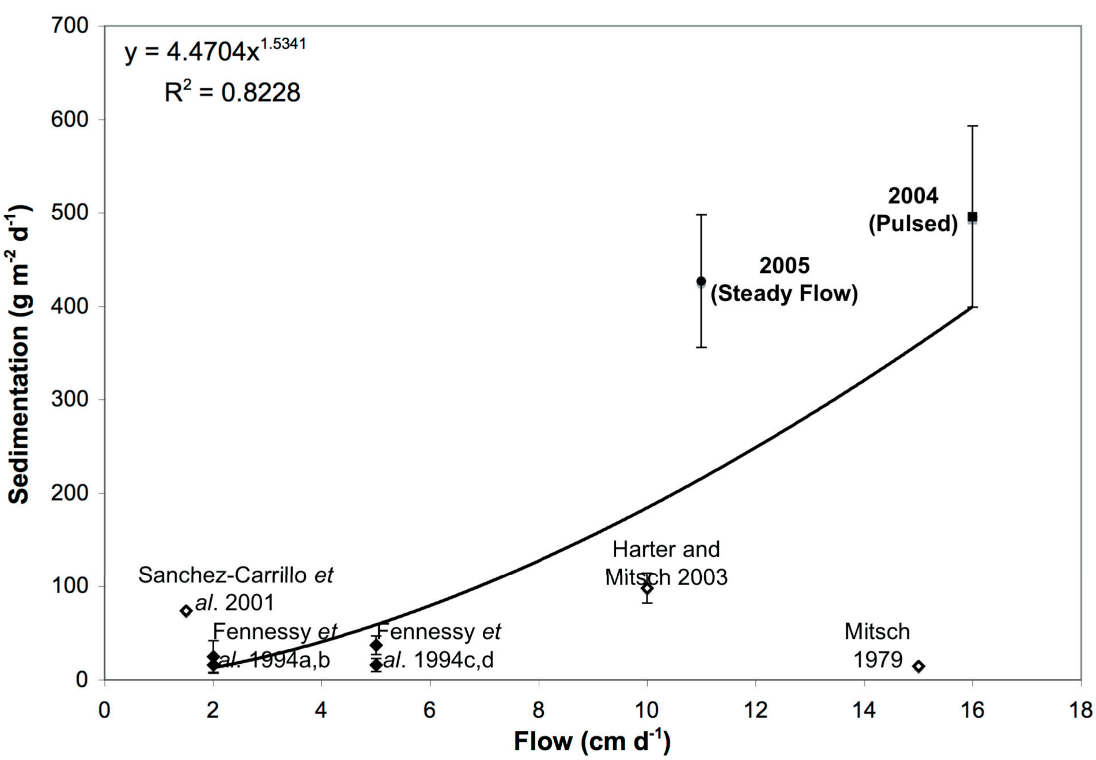

Figure 6. Sedimentation rates (g-dry wt m^{-2} d^{-1}) as a function of flow (cm d^{-1}) for several studies. Means are reported with standard error when possible. Regression line is for low flow (Fennessy et al. 1994) and high flow (this study) wetlands using the bottle trap method. Unfilled diamonds (◊) are for studies using other methods of estimating sedimentation rates or studies focused on wetlands other than riparian marshes.

2004 had a high sedimentation rate relative to imported TSS, which accounts for the higher mean S:I ratio for the pulsed year. This also indicates that many of the sediments contributing to the sedimentation rates are recycled from within the wetland. S:O was higher in the 2005 steady-flow year (S:O of 811) than in the 2004 pulsed year (S:O of 406), suggesting that there was a higher export in the pulsed year than the steady-flow year. This may be explained by slightly higher water flow entering, and, therefore, leaving the wetlands.

Sedimentation and nutrient dynamics

The Redfield ratio is the ratio of C:N:P that Redfield found in ocean plankton (Redfield, 1958); this ratio is often used to calculate excess or limiting nutrients within ecosystems. Redfield's ratio describes a C:N:P ratio in a balanced system at 41C:7.2N:1P (by weight). Carbon is generally not limiting in wetland systems, and this is evident in the ratios calculated from nutrient data; mean ratios for sediments were 77C:5N:1P in 2004 and 93C:5N:1P in 2005. The nitrogen to phosphorus ratio is more important. Mean N:P ratios calculated from nutrient data on collected sediments were 5:1, with no significant differences between the pulsed and steady-flow years (with a range of 3:1 to 7:1 in individual samples). These N:P ratios, suggest that nitrogen, rather than phosphorus, may be the limiting nutrient in the experimental wetlands. In an experiment conducted in 1996, Svengsouk and Mitsch (2001) reported nitrogen-limited conditions in the same experimental wetlands used in this study.

Organic matter plays an important part in nitrogen cycling, as many forms of nitrogen (i.e., ammonium) are bound to organic matter until decomposition or uptake by vegetation (Mitsch and Gosselink, 2000); thus, nitrogen concentrations are expected to be correlated to percent organic matter in sediments. In this study, there is a weak positive relationship between the percent organic matter and the percent nitrogen in collected sediments (Figure 7a; Pearson Correlation coefficient = 0.354, p = 0.001). Since rates of organic accumulation are low (total sediment accumulation was 14 ± 2 kg m^{-2} in 2004 and 11 ± 2 kg m^{-2} in 2005), nitrogen associated with these types of sediments are also accumulating in the soil at low rates. It is also possible that nitrogen in sediments is either absorbed by vegetation or quickly transformed, resulting in very short residence times as sediment-bound nitrogen.

Phosphorus transformations are dominated by sedimentary processes. When open binding sites are available, positively charged clay particles, or inorganic material, have a high potential for binding and storing negatively charged phosphorus ions (Mitsch and Gosselink, 2000). Phosphorus concentrations were expected to increase with the percent

Table 3. Monthly sediment budgets, including 4-month total and monthly mean suspended solid (TSS) loads for the inflow and outflow of the wetlands (g m^{-2} mo^{-1}), measured sedimentation (g m^{-2} mo^{-1}), and ratios sedimentation:inflow TSS (S:I) and sedimentation:outflow TSS (S:O) in 2004 pulsed and 2005 steady-flow years. Ratios are for the sum of the four-month budget components.

			2004 Pulsed		
	I TSS In (g m^{-2} mo^{-1})	O TSS Out (g m^{-2} mo^{-1})	S Sedimentation (g m^{-2} mo^{-1})	S:I	S:O
April	73	90	8726	119	97
May	56	40	13,238	236	332
June	79	62	23,308	296	374
July	15	29	44,602	2958	1527
Total	223	221	89,874		
Mean	56	55	22,468	403	406
			2005 Steady Flow		
	I TSS In (g m^{-2} mo^{-1})	O TSS Out (g m^{-2} mo^{-1})	S Sedimentation (g m^{-2} mo^{-1})	S:I	S:O
April	32	33	8167	252	250
May	21	18	10,626	510	589
June	14	13	19,986	1428	1585
July	29	15	24,722	858	1647
Total	96	78	63,501		
Mean	24	20	15,875	661	811

inorganic material that comprised sediments. Regression analyses did not support this hypothesis, showing that there was a significant inverse relationship between percent inorganic material and phosphorus concentrations (Figure 7b; Pearson Correlation coefficient = -0.297, p = 0.006). This may be because many of the trapped sediments were from allochthonous sources, which input freshly sheered sediment particles that are relatively void of nutrients.

Pulses seem to affect rates of nutrient sedimentation by distributing nutrients throughout the wetland, while there are decreasing rates of nutrient sedimentation from inflow to outflow during the steady-flow regime (Table 2). The pulsing is important in nutrient distribution, making them more available for transformations, vegetative uptake, and long-term storage over a larger area. The amount of nutrient sedimentation measured is considerably high in the experimental wetlands, with means of 834 ± 190 g-N m^{-2} yr^{-1} and 441 ± 130 g-N m^{-2} yr^{-1} of nitrogen deposited in 2004 and 2005, respectively. While these high numbers may be in part due to resuspension, nitrogen loading into wetlands at this rate surpasses the sustainable range of 10 to 40 g-N m^{-2} yr^{-1} suggested by Mitsch et al. (2000). Phosphorus sedimentation was considerably lower than nitrogen sedimentation, with means of 200 ± 40 g-P m^{-2} yr^{-1} and 100 ± 40 g-P m^{-2} yr^{-1} of phosphorus deposited for 2004 and 2005, respectively. These sedimentation rates for phosphorus also exceed the range of 0.5–5 g-P m^{-2} yr^{-1} of designated sustainable levels (Mitsch et al., 2000).

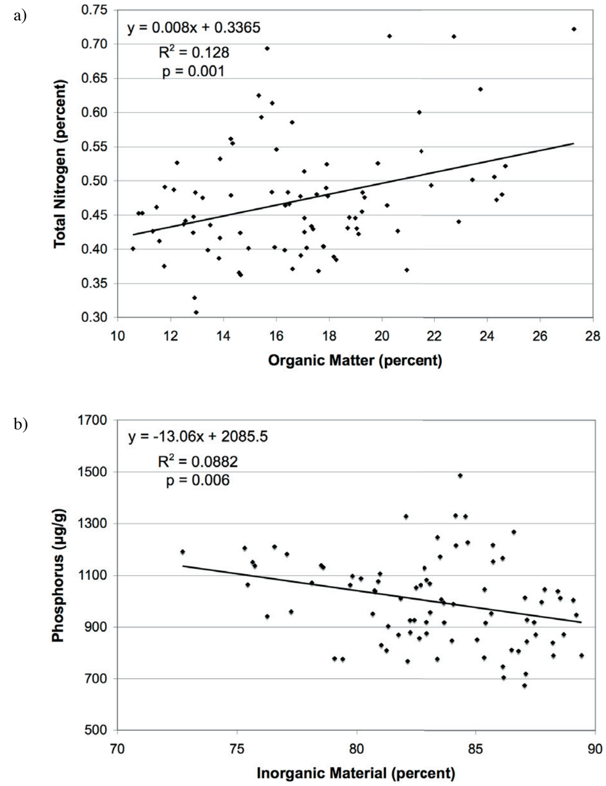

Figure 7. (a) Percent nitrogen as a function of percent organic matter and (b) phosphorus concentrations (μg/g) as a function of percent inorganic material in collected sediments. P-value determined using a Pearson Correlation.

Factors that influence sedimentation measurements

Resuspension

Sedimentation rates gathered in this study may have been influenced by resuspension. Harter and Mitsch (2003) estimated that approximately 36 kg m^{-2} yr^{-1} of sediments were resuspended due to wind, ice, and animal activity, while net sediment input and output were an order of magnitude lower. High amounts of resuspension most likely contributed to the high sedimentation rates calculated from this study.

Supporting evidence for resuspension comes from the gradual increase of sedimentation rates from April through June, with a sharp increase in sedimentation rates in July, a month after pulsing was completed (Figure 3). During pulsed months, water level within the wetland increased, resulting in higher water velocities and disrupted flow paths (i.e., water moves laterally instead of vertically), both of which encourage resuspension of sediments through substrate disturbance or erosion. High water velocities and disrupted flow paths seem to act together to increase the suspended sediment load and retain water within the wetland. After the pulse ends and a steady-flow regime is applied, the expanded wetland begins to contract and the large volume of sediment laden water retained during the pulse is slowly exported (Tockner et al., 2000). This results in a net export of sediments well after the pulse. Water velocities, however, most likely remain elevated during April, May, and June of 2004 due to the expansion–contraction process. It is not until July of the same year that sedimentation rates sharply increase, indicating that a period of residence time, with no dynamic fluctuations, is needed after a pulse to allow the wetland to equilibrate and suspended sediments to settle from the water column. Kleiss (1996) reported similar patterns of sedimentation after river pulsing in bottomland hardwood forests.

While water velocities and wetland expansion and contraction may have the greatest influence in resuspension rates of sediments, other studies have reported that wind is the most common cause of resuspension events (Cózar et al., 2005). Wind speed did not differ between the pulsed and steady-flow years in this study, with means of 3.3 m sec^{-1} in both 2004 and 2005, and wind direction did not substantially change between years. Temporal variations in sedimentation cannot be explained by resuspension due to wind; however, spatial variations in sedimentation may be due to wind resuspension. Cózar et al. (2005) reported that the frequency of resuspension events increases exponentially as water depths decrease; therefore, it is likely that shallow areas were affected by wind resuspension. Higher sedimentation rates reported in shallow vegetated areas in this study may be a result of more frequent wind resuspension (Figure 4).

Calcium precipitation

Under high temperatures and high pH typical conditions in many wetlands, calcium carbonate precipitation can occur. Algae play a large role in calcium precipitation directly, by influencing this process by driving high pH conditions through removal of CO_2 during photosynthesis, and indirectly, by incorporating calcium carbonate into their structure, which eventually (i.e., in death) contributes to sedimentation rates (Liptak 2000). Therefore, algal production in the wetlands was expected to influence calcium precipitation in the wetlands. By measuring gross primary productivity of the water column (data provided by Tuttle, 2005) and rates of calcium sedimentation during the study periods, a positive relationship between gross primary productivity and calcium sedimentation was established (Figure 5b; Pearson Correlation coefficient = 0.760, p = 0.047). Mean calcium sedimentation rates in this study ranged from 12–39 g-Ca m^{-2} d^{-1} during the pulsed year to 6–29 g-Ca m^{-2} d^{-1} during the steady-flow year. Liptak (2000) reported that mean net calcium carbonate deposition in the experimental wetlands at the Olentangy River Wetland Research Park occurred at a rate of 0.48 g-Ca m^{-2} d^{-1}, a rate far lower than reported here.

Macrophytes

Vegetation is generally reported to have an important effect on sedimentation, although its role has been disputed. Some studies have shown that sedimentation rates are highest in vegetated areas due to decreased water velocities and effective trapping of sediments by stems and roots (Peterson and Teal, 1996), while other studies have shown that vegetation deters sedimentation by re-directing flow towards open water areas (Fennessy et al., 1994). Brueske and Barrett (1994) reported that sedimentation rates were highest in vegetated areas under high flow conditions and highest in open water areas under low flow conditions. Finally, there are studies that report that vegetation has no effect on sedimentation rates (Steiger and Gurnell, 2002, Temmerman et al., 2003).

In this study, vegetation appeared to positively affect sedimentation rates, with the greatest sedimentation rates occurring in shallow, vegetated areas. This is also supported by the facts that sedimentation significantly increased as the growing season progressed (most likely due to more trapping of sediments by vegetation) and sedimentation rates were highest in shallow, vegetated areas. An indirect contribution of vegetation appears as increased organic (autochthonous) sedimentation rates during the growing season. This could be a result of resuspension and biodegradation of the previous year's vegetation as new vegetation is emerging.

Study limitations

The bottle trap method in this study was necessary to capture a sufficient amount of sediment for lab analysis and to provide an easily repeatable technique. It was equally as important for the traps to set for the full month because, even after a month, some traps did not catch enough sediment for full characterization analysis. In the future, the combination of bottle traps to catch sediments for analysis, horizon markers to more accurately estimate net sedimentation rates, and stable isotope characterization to determine the source of sediments, would help describe sedimentation pathways in wetlands.

Simulated flood pulses in this study did not necessarily exemplify sediment load characteristics of actual flood pulses. Flooding events generally result in bank erosion and higher suspended material loads (Knighton, 1998); therefore, river floodwaters naturally deposit more sediments into the riparian wetlands to which they are connected (Mitsch, 1979). Due to the experimental design of this study, simulated flooding often occurred outside flood conditions in the river; therefore, sedimentation rates, especially those for allochthonous inputs, were most likely lower than they would be during a natural pulse.

Soil development in riparian wetlands

Concentrations of nitrogen and phosphorus in wetland soils (i.e., substrate) were lower than those in trapped sediments. Mean concentrations for nitrogen and phosphorus were 0.35% and 810 μg/g for wetland soils compared to 0.47% and 1013 μg/g for trapped sediments. This suggests that the wetlands in this study continue to be nourished by fresh sediments from both river and internal sources. Differences in nutrient concentrations between trapped sediments and wetland soils may be due to vegetative uptake or chemical transformations in the reduced substrate. The process of introducing new sediments into the wetlands, especially those from allochthonous sources with high sorption potentials, is important for nutrient binding and transport to vegetative and microbial communities for removal.

Conclusions

Floodplain restoration is essential for the restoration of rivers. Re-establishing a natural hydrologic regime and the exchange of materials between these systems involves both temporarily flooded floodplains and more frequently flooded riparian wetlands. Restoration can be accomplished by reconnecting wetlands to the river by removing or breeching levees, or creating swales that allow for river water to enter adjacent wetlands. The exchange of nutrients and materials between the river and riparian wetland systems drives nutrient cycling and biotic establishment. Sediments and organic material provided by natural river pulsing are especially important subsidies to riparian wetlands during early spring months, while a natural hydrologic regime is essential for recharging the riparian wetland with new sediments, flushing old sediments from the wetlands, and providing the river with a source of organic matter. A hydrologic regime with occasional pulsing contributes to the function of a healthy and dynamic riparian wetland system.

This study provided the following conclusions related to sedimentation and sedimentation measurement techniques in freshwater alluvial wetlands:

1. The bottle trap method over-estimates sedimentation rates, but can be used effectively for comparative studies.

2. Sedimentation rates normalized for differences between years were similar in pulsing and steady-flow years.

3. Short-term (monthly) patterns of sedimentation were different between pulsing and steady-flow years, regardless of whether rates were normalized.

4. Allochthonous inputs account for the majority of introduced sediments in these riparian wetlands with autochthonous inputs becoming more important in the growing season and in steady-flow conditions.

5. Basin morphology and vegetation may act together to affect sedimentation rates.

6. A pulsing regime aids in nutrient distribution by spreading sediments, new and resuspended, throughout the wetland.

7. Flood pulses result in the exchange of new, more nutrient-rich sediments for old sediments, subsidizing both wetland and river processes.

8. Optimizing the exchange of sediments between the river and its floodplain is essential for floodplain and river restoration.

9. Specific to restoration of large deltas, such as that in Louisiana, sedimentation, wetland restoration, and deltaic accretion may be optimized by allowing ample time between periods of pulsing diversions.

References

American Public Health Association (APHA), American Water Works Association (AWWA) and Water Environment Federation (WEF). 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition. United Book Press, Inc., 6Baltimore, Maryland.

Association of Analytical Communities (AOAC) International. 2002. Official Method 990.03: Protein (crude) in Animal Feed Combustion Method (Dumas method). In: W. Horwitz (ed.), Official Methods of Analysis, 17th edition.

Baker, D. B., R. P. Richards, T. T. Loftus and J. W. Kramer. 2004. A new flashiness index: characteristics and applications to Midwestern rivers and streams. Journal of the American Water Resources Association 40: 503-522.

Boumans, R. M. and J. W. Day, Jr. 1994. Effects of two Louisiana marsh management plans on water and materials flux and short-term sedimentation. Wetlands 14: 247-261.

Braskerud, B. C. 2001. The influence of vegetation on sedimentation and resuspension of soil particles in small constructed wetlands. Journal of Environmental Quality 30: 1447-1457.

Britsch, L. D. and J. B. Dunbar. 1993. Land loss rates: Louisiana coastal plain. Journal of Coastal Research 9: 324-338.

Brinson, M. M. 1977. Decomposition and nutrient exchange of litter in an alluvial swamp forest. Ecology 58: 601-609.

Brinson, M. M., A. E. Lugo and S. Brown. 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics 12: 123-161.

Brown, R. G. 1985. Effects of an urban wetland on sediment and nutrient loads in runoff. Wetlands 4: 147-158.

Brueske, C. C. and G. W. Barrett. 1994. Effects of vegetation and hydrologic load on sedimentation patterns in experimental wetland ecosystems. Ecological Engineering 3: 429-447.

Callaway, J. C., J. A. Nyman and R. D. DeLaune. 1996. Sediment accretion in coastal wetlands: a review and simulation model of processes. Current Topics in Wetland Biogeochemistry 2: 2-23.

Canfield, D. E. 1997. The geochemistry of river particulates from the Continental USA: major elements. Geochimica et Cosmochimica Acta 61: 3349-3365.

Comin, A. F., J. A. Romero, V. Astorga and C. Garcia. 1997. Nitrogen removal and cycling in restored wetlands used as filters of nutrients for agricultural runoff. Water Science and Technology 35: 255-261.

Connors, L. M., E. Kiviat, P. M. Groffman and R. S. Ostfeld. 2000. Muskrat (Ondatra zibethicus) disturbance to vegetation and potential net nitrogen mineralization and nitrification rates in a freshwater tidal marsh. American Midland Naturalist 143: 53-63.

Cózar, A., J. A. Gálvez, V. Hull, C. M. García and S. A. Loiselle. 2005. Sediment resuspension by wind in a shallow lake of Esteros del Iberá (Argentina): a model based on turbidimetry. Ecological Modelling 186: 63-76.

Craft, C. B. and W. P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20: 323-332.

Dahl, T. E. 1990. Wetlands Losses in the United States, 1780s to 1980s. U.S. Department of the Interior, Fish and Wildlife Service, Washington D.C., USA.

Darke, A. K. and J. P. Megonigal. 2003. Control of sediment deposition rates in two mid-Atlantic Coast tidal freshwater wetlands. Estuarine, Coastal and Shelf Science 57: 255-268.

Davide, V., M. Pardos, J. Diserens, G. Ugazio, R. Thomas and J. Dominik. 2003. Characterization of bed sediments and suspension of the river Po (Italy) during normal and high flow conditions. Water Research 37: 2847-2864.

Davies, B. E. 1974. Loss on ignition as an estimate of soil organic matter. Soil Science Society of America Proceedings 38: 150-151.

Day, J. W., Jr., A. Y. Arancibia, W. J. Mitsch, A. L. Lara-Dominguez, J. N. Day, Jae-Young Ko, R. Lane, J. Lindsey and D. Z. Lomeli. 2003a. Using Ecotechnology to address water quality and wetland habitat loss problems in the Mississippi basin: a hierarchical approach. Biotechnology Advances 22: 135–159.

Day, J. W., J. Ko, J. Cable, J. N. Day, B. Fry, E. Hyfield, D. Justic, P. Kemp, R. Lane, H. Mashriqui, E. Reyes, S. Rick, G. Snedden, E. Swenson, P. Templet, R. Twilley, K. Wheelock and B. Wissel. 2003b. Pulses: the importance of pulsed physical events for Louisiana floodplains and watershed management. The First Interagency Conference on Research in the Watersheds, 27-30 October 2003, Benson, AZ, USA.

Day, J. W. Jr., J. Ko, J. Rybczyk, D. Sabins, R. Bean, G. Berthelot, C. Brantley, L. Cardoch, W. Conner, J. N. Day, A. J. Englande, S. Feagley, E. Hyfield, R. Lane, J. Lindsey, J. Mistich, E. Reyes and R. Twilley. 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: a review. Ocean and Coastal Management 47: 671-691.

Delong, M. D., J. H. Thorp, K. S. Greenwood and M. C. Miller. 2001. Responses of consumers and food resources to a high magnitude, unpredicted flood in the Upper Mississippi River Basin. Regulated Rivers: Research and Management 17: 217-234.

DeLaune, R. D., A. Jugsujinda, G. W. Peterson and W. H. Patrick, Jr. 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana estuary. Estuarine, Coastal and Shelf Science 58: 653-662.

Fennessy, M. S., C. C. Brueske and W. J. Mitsch. 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. Ecological Engineering 3: 409-428.

Forster, J. 1995. Organic Carbon. In: Alef, K. and P. Nannipieri (eds.), Methods in Applied Soil Microbiology and Biochemistry. Academic Press Inc., San Diego, CA, USA: 59-65.

Gala, D. L., L. H. Fredrickson, D. D. Humburg, K. J. Bataille, J. R. Bodie, J. Dohrenwend, G. T. Gelwicks, J. E. Havel, D. L. Helmers, J. B. Hooker, J. R. Jones, M. F. Knowlton, J. Kubisiak, J. Mazourek, A. C. McColpin, R. B. Renken, and R. D. Semlitsch. 1998. Flooding to restore connectivity of regulated, large-river wetlands. BioScience 48: 721-733.

Gardener, W. D. 1980. Sediment trap dynamics and calibration: a laboratory evaluation. Journal of Marine Research 38: 17-39.

Gichuki, J., L. Triest and F. Dehairs. 2001. The use of stable carbon isotopes as tracers of ecosystem functioning in contrasting wetland ecosystems of Lake Victoria, Kenya. Hydrobiologia 458: 91-97.

Hakanson, L. and M. Jansson. 1983. Principles of Lake Sedimentology. Springer, New York, NY, USA.

Harter, S. K. 1999. Patterns of short-term sedimentation in a freshwater created marsh. MS Thesis. School of Natural Resources, The Ohio State University, Columbus, OH, USA.

Harter, S. K. and W. J. Mitsch. 2003. Patterns of short-term sedimentation in a freshwater created marsh. Journal of Environmental Quality 32: 325-334.

Hatton, R. S., R. D. DeLaune and W. H. Patrick, Jr. 1983. Sedimentation, accretion, and subsidence in marshes of Barataria Basin, Louisiana. Limnology and Oceanography 28: 494-502.

Heimann, D. C. and M. J. Roell. 2000. Sediment loads and accumulation in a small riparian wetland system in northern Missouri. Wetlands 20: 219-231.

Horne, A. J. and C. R. Goldman. 1994. Limnology, Second Edition. McGraw Hill, Inc., New York, NY, USA.

Hupp, C. R. and E. E. Morris. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. Wetlands 10: 107-124.

Junk, W. J., P. B. Bayley and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. In: D. P. Dodge (ed.), Proceedings of the International Large River Symposium. Canadian Special Publications of Fisheries and Aquatic Sciences 106. pp. 110-127.

Kang, H., C. Freeman, D. Lee and W. J. Mitsch. 1998. Enzyme activities in constructed wetlands: Implication for water quality amelioration. Hydrobiologia 368: 231-235.

Kadlec, R. H. and R. L. Knight. 1996. Suspended solids. In: Treatment Wetlands. Lewis Publishing, Boca Raton, FL. p. 315–339.

Kendall, C., S. R. Silva and V. J. Kelly. 2001. Carbon and nitrogen isotopic compositions of particulate organic matter in four large river systems across the United States. Hydrological Processes 15: 1301-1346.

Kleiss, B. A. 1996. Sediment retention in a bottomland hardwood wetland in eastern Arkansas. Wetlands 16: 321-333.

Knighton, D. 1998. Fluvial Forms and Processes: A New Perspective. Oxford University Press, New York, NY, USA.

Knowlton, M. F., C. Cuvellier and J. R. Jones. 2002. Initial performance of a high capacity surface-flow treatment wetland. Wetlands 22: 522-527.

Konen, M. E., P. M. Jacobs, C. L. Burras, B. J. Talaga and J. A. Mason. 2002. Equations for predicting soil organic carbon using loss-on-ignition for North Central

U.S. soils. Soil Science Society of America Journal 66: 1878-1881.

Koreny, J. S., W. J. Mitsch, E. S. Bair and X. Wu. 1999. Regional and local hydrology of a created riparian wetland system. Wetlands 19: 182-193.

Lane, R. R., J. W. Day, Jr. and B. Thibodeaux. 1999. Water quality analysis of a freshwater diversion at Caernarvon, Louisiana. Estuaries 22: 327-336.

Lane, R. R., J. W. Day, D. Justic, E. Reyes, B. Marx, J. N. Day and E. Hyfield. 2004. Changes in stoichiometric Si, N and P ratios of Mississippi River water diverted through coastal wetlands to the Gulf of Mexico. Estuarine, Costal and Shelf Science 60: 1-10.

Liptak, M. A. 2000. Water column productivity, calcite precipitation, and phosphorus dynamics in freshwater marshes. PhD Dissertation. The Ohio State University, Columbus, OH, USA.

Loucks, O. L. 1990. Restoration of the pulse control function of wetlands and its relationship to water quality objectives. In: Kusler, J. A. and M. E. Kentula (eds.), Wetland Creation and Restoration: The Status of Science. U.S. EPA, Corvallis, OR, USA. pp. 467-477.

Mann, C. J. and R. G. Wetzel. 2000. Hydrology of an impounded lotic wetland – wetland sediment characteristics. Wetlands 20: 23-32.

Martin, J. F., E. Reyes, G. P. Kemp, H. Mashriqui and J. W. Day, Jr. 2002. Lanscape modeling of the Mississippi Delta. BioScience 52: 357-365.

Metzker, K. and W. J. Mitsch. 1997. Modelling self-design of the aquatic community in a newly created freshwater wetland. Ecological Modelling 100: 61-86.

Middleton, B. 2000. Hydrochory, seed banks, and regeneration dynamics along the landscape boundries of a forested wetland. Plant Ecology 146: 169-184.

Middleton, B. 2002. The flood pulse concept in wetland restoration. In: Middleton, B. (ed.), Flood Pulsing in Wetlands: Restoring the Natural Hydrological Balance. John Wiley and Sons, Inc., New York, NY, USA.

Mitsch, W. J. 1979. Interactions between a riparian swamp and a river in southern Illinois. In: J. McCormick and R. Johnson (eds.), Strategies for Protection and Management of Floodplain Wetlands and other Riparian Ecosystems. USDA Forest Service General Technical Report WO-12, Washington, DC. pp. 63-72.

Mitsch, W. J. 1984. Seasonal patterns of a cypress dome pond in Florida. In: K. C. Ewel and H. T. Odum (eds.), Cypress Swamps. University Presses of Florida, Gainsville, pp. 25-33.

Mitsch, W. J. 1995. Restoration of our lakes and rivers with wetlands – an important application of ecological engineering. Water Science and Technology 31: 167-177.

Mitsch, W. J., C. L. Dorge and J. R. Wiemhoff. 1979. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60: 1116-1124.

Mitsch, W. J., X. Wu, R. W. Nairn, P. E. Weihe, N. Wang, R. Deal and C. E. Boucher. 1998. Creating and restoring wetlands: a whole-ecosystem experiment in self-design. Bioscience 48: 1019-1030.

Mitsch, W. J. and J. G. Gosselink. 2000. Wetlands, Third Edition. John Wiley and Sons, Inc., New York, NY, USA.

Mitsch, W. J., A. J. Horne and R. W. Nairn. 2000. Nitrogen and phosphorus retention in wetlands – ecological approaches to solving excess nutrient problems. Ecological Engineering 14: 1-7.

Mitsch W. J., J. W. Day, Jr., J. W. Gilliam, P. M. Groffman, D. L. Hey, G. W. Randall and N. Wang. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: strategies to counter a persistent ecological problem. BioScience 51: 373– 88.

Mitsch, W. J. and J. W. Day, Jr. 2004. Thinking big with whole-system studies and ecosystem restoration – a legacy of H.T. Odum. Ecological Modelling 178: 133-155.

Mitsch, W. J., J. W. Day, Jr., L. Zhang and R. R. Lane. 2005a. Nitrate-nitrogen retention in wetlands in the Mississippi River Basin. Ecological Engineering 24: 267-278.

Mitsch, W. J., N. Wang, L. Zhang, R. Deal, X. Wu and A. Zuwerink. 2005b. Using ecological indicators in a whole-ecosystem wetland experiment. In: S. E. Jørgensen, F. L. Xu and R. Costanza (eds.), Handbook of Ecological Indicators for Assessment of Ecosystem Health. CRC Press, Boca Raton, FL. pp. 211-236.

Mitsch, W. J., L. Zhang, C. J. Anderson, A. Altor and M. Hernandez. 2005c. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecological Engineering 25: 510-527.

Mitsch, W. J. and J. W. Day, Jr. 2006. Restoration of wetlands in the Mississippi-Ohio-Missouri (MOM) River Basin: experience and needed research. Ecological Engineering 26: 55-69.

Nairn, R. W. and W. J. Mitsch. 2000. Phosphorus removal in created wetland ponds receiving river overflow. Ecological Engineering 14: 107-126.

Neumann, T., A. Stögbauer, E. Walpersdorf, D. Stüben and H. Kunzendorf. 2002. Stable isotopes in recent sediments of Lake Arendsee, NE Germany: response to eutrophication and remediation measures. Paleogeography, Paleoclimatology, Paleoecology 178: 75-90.

Odum, W. E., E. P. Odum and H. T. Odum. 1995. Nature's pulsing paradigm. Estuaries 18: 547-555.

Parsons, M., C. A. McLoughlin, K. A. Kotschy, K. H. Rogers and M. W. Rountree. 2005. The effects of extreme floods on the biophysical heterogeneity of river landscapes. Frontiers in Ecology and the Environment 3: 487-494.

Penland, S., L. Wayne, L. D. Britsch, S. J. Williams, A. D. Beall and V. C. Butterworth. 2000. Process classification of coastal land loss between 1932 and 1990 in the Mississippi River delta plain, southeastern Louisiana. USGS Open-File Report Number 2000-418.

Peterson, S. B. and J. M. Teal. 1996. The role of plants in ecologically engineered wastewater treatment systems. Ecological Engineering 6: 137-148.

Rabalais, N. N., R. E. Turner, D. Justic, Q. Dortch and W. J. Wiseman, Jr. 1999. Characterization of hypoxia: Topic 1. Report for the Integrated Assessment of Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series Number 15. NOAA Coastal Ocean Program, Silver Spring, MD.

Reddy, K. R. and W. H. Patrick, Jr. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. Soil Biology and Biochemistry 7: 87-94.

Redfield, A. C. 1958. The biological control of chemical factors in the environment. American Scientist 46: 206-226.

Reyes, E., R. R. Lane and J. W. Day. 2003. Watershed analysis of pulsing freshwater events using landscape modeling in Coastal Louisiana. The First Interagency Conference on Research in the Watersheds, 27-30 October 2003, Benson, AZ, USA.

Robertson, A. I., P. Bacon and G. Heagney. 2001. The responses of floodplain primary productivity to flood frequency and timing. Journal of Applied Ecology 28: 126-136.

Sánchez-Carrillo, S., M. Álvarez-Cobelas and D. G. Angeler. 2001. Sedimentation in the semi-arid freshwater wetland Las Tablas de Daimiel (Spain). Wetlands 21: 112-124.

Scavia, D., N. N. Rabalais, R. E. Turner, D. Justic and W. J. Wiseman, Jr. 2003. Predicting the response of Gulf of Mexico hypoxia to variations in Mississippi River nitrogen load. Limnology and Oceanography 48: 951-956.

Schumm, S. A. and M. A. Stevens. 1973. Abrasion in place: a mechanism for rounding and size reduction of coarse sediments in rivers. Geology 1: 37-40.

Selbo, S. M and A. A. Snow. 2004. The potential for hybridization between Typha angustifolia and T. latafolia in a constructed wetland. Aquatic Botany 78: 361-369.

Sparks, R. E., J. C. Nelson and Y. Yin. 1998. Naturalization of the flood regime in regulated rivers. Bioscience 48: 706-720.

Spieles, D. J. and W. J. Mitsch. 2000a. The effects of season and hydrologic and chemical loading on nitrate retention in constructed wetlands: a comparison of low and high nutrient riverine systems. Ecological Engineering 14: 77-91.

Spieles, D. J. and W. J. Mitsch. 2000b. Macroinvertebrate community structure in high-and low-nutrient constructed wetlands. Wetlands 20: 716-729.

Steiger, J. and A. M. Gurnell. 2002. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. Geomorphology 49: 1-23.

Steiger, J., A. M. Gurnell and J. M. Goodson. 2003. Quantifying and characterizing contemporary riparian sedimentation. River Research and Applications 19: 335-352.

Svengsouk, L. J. and W. J. Mitsch. 2001. Dynamics of mixtures of Typha latifolia and Schoenoplectus tabernaemontani in nutrient-enrichment wetland experiments. American Midland Naturalist 145: 309-324.

Temmerman, S., G. Govers, S. Wartel and P. Meire. 2003. Spatial and temporal factors controlling short-term sedimentation in a salt and freshwater tidal marsh, Scheldt Estuary, Belgium, SW Netherlands. Earth Surfact Processes and Landforms 28: 739-755.

Tockner, K., F. Malard and J. V. Ward. 2000. An extension of the flood pulse concept. Hydrological Processes 14: 2861-2883.

Tockner, K. and J. A. Stanford. 2002. Riverine flood plains: present state and future trends. Environmental Conservation 29: 308-330.

Toth, L. A., J. W. Koebel, Jr., A. G. Warne and J. Chamberlain. 2002. Implications of reestablishing prolonged flood pulse characteristics of the Kissimmee River and floodplain ecosystem. In: Middleton, B. (ed.), Flood Pulsing in Wetlands: Restoring the Natural Hydrological Balance. John Wiley and Sons, Inc., New York, NY, USA. pp. 191-219.

Tsunis, I. K., K. L. Prescott and H. Shen. 1998. Modelling of phosphorus and suspended solids in Cootes Paradise Marsh. Ecological Modelling 114: 1-17.

Tuttle, C. L. 2005. The effects of hydrologic pulsing on aquatic metabolism in created riparian wetlands. MS Thesis. Environmental Science Graduate Program, The Ohio State University, Columbus, OH, USA.

Ulbrich, K., R. Marsula, F. Jeltsch, H. Hofmann and C. Wissel. 1997. Modelling the ecological impact of contaminated river sediments on wetlands. Ecological Modelling 94: 221-230.

U.S. Environmental Protection Agency (EPA). 1994. Test Methods for Evaluating Solid Waste, SW-846, 3rd Edition. Office of Solid Waste and Emergency Response, Washington, D.C., November 1986; Update II, September 1994.

Vannote, R. L., G. M. Minshall, K. W. Cummins, J. R. Sedell and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Verhoeven, J. T. A., D. F. Whigham, R. Van Logtestijn and J. O'Neill. 2001. A comparative study of nitrogen and phosphorus cycling in tidal and non-tidal riverine wetlands. Wetlands 21: 210-222.

Wardrop, D. H. and R. P. Brooks. 1998. The occurrence and impact of sedimentation in central Pennsylvania wetlands. Environmental Monitoring and Assessment 51: 119-130.

Wu, X, and W. J. Mitsch. 1998. Spatial and temporal patterns of algae in newly constructed freshwater wetlands. Wetlands 18: 9-20.

Zhang, L. and W. J. Mitsch. 2005. Modelling of hydrological budgets of flow-through created wetlands: An integrated system approach. Environmental Software & Modelling 20: 935-946.

Ziegler, S. E. and M. L. Fogel. 2003. Seasonal and diel relationships between the isotopic compositions of dissolved and particulate organic matter in freshwater ecosystems. Biogeochemistry 64: 25-52.

Zierholz, C., I. P. Prosser, P. J. Fogarty and P. Rustomji. 2001. In-stream wetlands and their significance for channel filling and the catchment sediment budget, Jugiong Creek, New South Wales. Geomorphology 38: 221-235.